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Impact on mortality of biomass combustion from wildfires in Spain: A regional analysis

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HIGHLIGHTS

- Daily mean PM concentrations were higher on DBA than on DNBA.
- PM_{10} was associated with higher daily mortality on DBA in regions where wildfires were most frequent.
- The increase in PM is linked to a significant IRR of mortality in Spain.
- Wildfires are likely to become increasingly frequent in the context of climate change makes this type of analysis necessary

GRAPHICAL ABSTRACT

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ABSTRACT

Studies that analyse the impact on mortality of particulate matter (PM) produced by biomass combustion from wildfires mostly focus on a single city or on cities in different countries, with very few concentrating on one country as a whole. Accordingly, the aim of this paper was to analyse the impact that PM has on daily mortality in Spain on days with biomass combustion from wildfires.

To analyse natural PM advections the Ministry of Agriculture and Fishing, Food & Environment divides Spain into 9 geographical regions. One province representative of each region for was selected analysis purposes, with provincial daily natural-cause mortality across the period 2004–2009 as the dependent variable, and daily mean PM concentrations in the provincial capital as the independent variable. We controlled for the effect of other chemical pollutants (NO_2 and O_3), maximum daily temperature on heat-wave days, day of the week, trends, seasonalities and the autoregressive nature of the series, using generalised linear models with the Poisson regression link to calculate relative risks (RRs) and the increase in RR (IRR) of PM-related mortality. The analysis was performed for days with and without biomass advections (DBA and DNBA respectively), with a breakdown by year, summer, and the remainder of the year (i.e., excluding summer).

The results indicated that daily mean PM concentrations were higher on DBA than on DNBA, with statistically significant differences in most provinces. Furthermore, PM_{10} was associated with higher daily mortality on DBA in regions where wildfires were most frequent, but not in the remaining provinces. This translated as an IRR per 10 $\mu g/m^3$ of PM of 7.93 (2.36–13.81) in the North-west, 3.76 (1.36–6.22) in the Centre and 4.46 (2.99–5.94) in the South-west, values which in all cases were statistically higher than those obtained on DNBA.

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The increase in PM caused by biomass advections from wildfires is linked to a significant IRR of mortality in Spain. Hence, the fact that wildfires are likely to become increasingly frequent in the context of climate change makes this type of analysis particularly necessary.

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1. Introduction

The main health impacts associated with particulate matter (PM) occur in densely populated urban areas (Ortiz et al., 2017) where the principal component is anthropic in nature (Querol et al., 2012), i.e., if one takes an urban area in southern Europe by way of example, 80% of such particulate and aerosol emissions will be seen to be linked to anthropogenic activities, while the remaining 20% continue to have a component of natural origin, stemming from advections of desert dust (Pey et al., 2013), sea spray (O'Dowd and de Leeuw, 2007), volcanic emissions (Von Glasow et al., 2009), and aerosols from wildfires (Barbosa et al., 2009). This anthropic origin of PM in cities is in contrast to the pattern observed at a global level, where 98% of all emissions are of natural origin (Guieré and Querol, 2010), fundamentally consisting of aerosols of marine origin (84–85%) and dust of mineral origin (13%).

The natural origin of PM in urban areas is of great importance, since this pollution is in addition to that which already exists in cities. In Spain, this inflow of PM serves to increase annual mean emission levels to 4% (Viana et al., 2014), basically in particles having a diameter of under 10 micra (PM_{10}). Although such values might on average seem of little significance, these contributions nevertheless tend to come in episodic periods of short duration which usually result in notable exceedances of the thresholds stipulated by European directives (Official Journal of the European Union, 2008) and the air quality guideline values established by the World Health Organization for protection of health (WHO, 2006), essentially in terms of the wide margin whereby PM_{10} and $PM_{2.5}$ levels are surpassed (Dennekamp and Abramson, 2011; Finlay et al., 2012).

The contribution to PM levels by sources of natural origin in southern Europe is fundamentally due to Saharan dust intrusions and PM advections from wildfires. The health effects of Saharan dust have been widely studied in Spain, with its impact being analysed in terms both of daily mortality (Jiménez et al., 2010; Tobías et al., 2011; Pérez et al., 2012; Díaz et al., 2012; Díaz et al., 2017) and of hospital admissions (Reyes et al., 2014). Far fewer studies have, however, analysed the health impact of PM produced by biomass combustion from wildfires (Finlay et al., 2012). At an international level, these studies have generally focused on hospital admissions due to ensuing respiratory problems (Mirabelli et al., 2009; Ovadnevaite et al., 2006) or exacerbations of previous respiratory diseases (Lee et al., 2009; Martin et al., 2013). Recently, exposure to forest fire smoke has also been linked to cardiac diseases (Dennekamp et al., 2015; Weichenthal et al., 2017).

From the standpoint of this PM's impact on mortality, existing studies are few in number, local in scope, and somewhat inconclusive. While a Denver-based study reported no association between these events and increases in daily mortality (Vedal and Dutton, 2006), others such as that undertaken in Athens, showed that so-called medium-sized fires do indeed have an influence on increases in daily mortality (Analitis et al., 2012). Another study conducted in the city of Madrid, which analysed the impact had by PM advections from biomass combustion on daily all-cause, circulatory-cause and respiratory-cause mortality (Linares et al., 2015), reported the existence of a shift in the PM-related mortality pattern on days with as opposed to those without advection, characterised by an increase in the relative risk (RR) of PM_{10}-related mortality, essentially in the over-75 age group. Similar results were obtained in a study undertaken in 10 southern European cities (Faustini et al., 2015), which found that "smoke is associated with increased cardiovascular mortality in urban residents, and PM_{10} on smoky days has a larger effect on cardiovascular and respiratory mortality than on other days".

As stated above, though few in number, there are some studies which, at a city level, link the effect that locally occurring wildfires have on daily mortality in nearby towns, but there are no studies, such as the one reported here, which analyse this effect at the level of an entire country like Spain, with a high number of fires every year and a mean annual affected surface area of 133,000 ha between 2006 and 2015, according to the latest data available from the Spanish Ministry of Agriculture and Fishing, Food & Environment. In 2016 alone, there were 517 wildfires in Spain (MAPAMA, 2016). If one then adds the fact that the predicted meteorological conditions associated with climate change in Spain are likely to further favour the development of wildfires (Moreno et al., 2015), regional studies such as this must be seen as indispensable.

2. Material and methods

2.1. Determination of days with biomass advections (DBA) and days without biomass advections from wildfires (DNBA)

To determine DBA during the period 2004–2009, we used information supplied by the Ministry of Agriculture and Fishing, Food & Environment (*Ministerio de Agricultura y Pesca, Alimentación y Medio Ambiente/MAPAMA* http://www.mapama.gob.es/es/calidad-y-evaluacion-ambiental/temas/atmosfera-y-calidad-del-aire/calidad-del-aire/evaluacion-datos/fuentes-naturales/default.aspx), which divides Spain into the following 9 main areas: North; North-east; North-west; Centre; South-west; South-east; Levant; Balearic Isles; and Canary Islands (Fig. 1). These areas were selected with the aim to cover the main different climatic and geographic conditions in Spain. The existence of regional-background monitoring stations in each region has been an additional factor to select them. As it has been explained in the manuscript, the time series of PM registered in this type of stations have been used for the identification of contributions of biomass burning emissions in Spain (Querol et al., 2010). Biomass burning events occurring in each main area were identified, based on the output from the NAAPS (Navy Aerosol Analysis and Prediction System) Global Aerosol Model. The NAAPS is a global aerosol transport model maintained operationally by the U.S. Navy for predicting the distribution of tropospheric aerosols (https://www.nrlmry.navy.mil/aerosol/). The emissions, transport and sinks of smoke from biomass burning, among other aerosol sources, are simulated as forecast products, and are thereafter corrected from satellite Aerosol Optical Depth measurements (Reid et al., 2009). For model initialisation, smoke emissions are estimated on the basis of geostationary satellite and MODIS fire products. The MODIS products build on heritage algorithms for operational fire monitoring used with the GOES and AVHRR sensors, and provide information on the location of a fire, its emitted energy, the flaming and smouldering ratio, and an estimate of area burnt (Giglio et al., 2003). The resulting product is then fed into the NAAPS for transport and removal. NAAPS uses global meteorological analyses and forecast fields from the Navy's 0.5° × 0.5° Operational Global Analysis and Prediction System (NOGAPS) on a 1° × 1° global grid at 24 vertical levels reaching 100 hPa. Four times daily, the NOGAPS weather forecast model provides dynamic and surface fields to the NAAPS at 6-hour intervals for a six-day forecast period. Transport is calculated using a 5th-order Lagrange

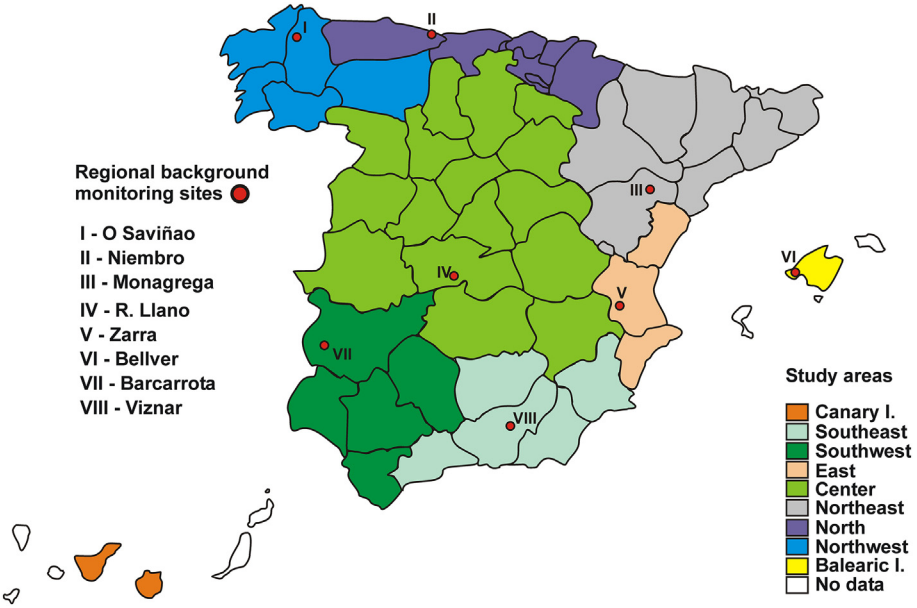

Fig. 1. Geographical location of the 9 main areas in Spain.

scheme with calculated departure points, while horizontal and vertical diffusion are calculated with a finite element scheme.

The impact of biomass burning smoke emissions on the PM levels registered in the study areas during each episodic day was then validated, in accordance with European Commission recommendations (Commission Staff Working Paper, 2011). Time series of daily PM_{10} data from eight, regional, background air-quality monitoring sites were compiled and analysed to evaluate the impact of biomass burning contributions on PM levels across the 6-year study period (Fig. 1).

Whereas seven out of the eight stations are members of the Co-operative Programme for Monitoring and Evaluation of Long-Range Transmission of Air Pollutants in Europe (EMEP), Monagrega forms part of the ENDESA (*Empresa Nacional de Electricidad S.A.*) Air Quality Network. These stations are located in rural areas, at distances of over 10 km from built-up areas and other major anthropogenic sources of air pollutants. They monitor background air-quality levels that are representative of a given area surrounding the station, ranging from 25 to 150 km, depending on the local topographical features (EEA, 1999). Hence, increases in PM_{10} levels at these stations during potential biomass burning episodes confirmed the occurrence of these events. Other studies have also used increases in regional background PM levels to validate satellite data on fires (Van Donkelaar et al., 2012).

The *MAPAMA* also provided information on days with Saharan dust contributions, for the above 9 main areas across the period 2004–2009. Details of the procedure used to identify the occurrence and duration of Saharan dust days can be found in Díaz et al. (2017).

2.2. Study variables

In each of the 9 regions into which Spain is divided, a representative province was selected in accordance with the availability and quality of the respective air pollution and mortality data. Chemical air pollution data, as supplied by the MAPAMA, corresponded to the daily mean values of each of the pollutants registered at the monitoring stations in each capital province across the period 2004–2009. The monitors used were mostly urban type, however when the data serie was not enough long, we used data from background monitors.

The independent variable was daily mean PM_{10} concentrations measured in $\mu g/m^3$. Only the Provinces of Madrid, Las Palmas and Santa Cruz de Tenerife had $PM_{2,5}$ measures. In these 3 cases, these concentrations were also included in the analysis.

As control variables, we used the mean daily NO_2 and O_3 concentrations obtained from the same source as those corresponding to PM, in provinces for which such data were available.

Given that the highest proportion of wildfires occurs in the summer period and on days with especially high temperatures, we controlled for the effect of maximum daily temperature in heat waves. For the definition of heat-wave temperature, we used the criterion defined in previous studies (Díaz et al., 2015). Maximum and minimum daily temperature data for each provincial capital were furnished by the State Meteorological Agency (*Agencia Estatal de Meteorología/AEMET*).

The dependent variable was daily mortality due to all causes except accidents (International Classification of Diseases 10th Revision (ICD-10): A00-R99), which occurred in towns of over 10,000 inhabitants by province across the period 2004–2009; except for the city of Madrid where the number corresponds to only metropolitan Madrid. With a breakdown by geographical location into the above-mentioned 9 areas (North, North-east, North-west, Centre, South-west, South-east, Levant, Balearic Isles and Canary Islands), as supplied by the National Statistics Institute (*Instituto Nacional de Estadística/INE*).

2.3. Methodology of analysis

We conducted a longitudinal ecological time-series study to analyse the relationship between daily mortality and PM_{10} levels in the period from 1 January 2004 to 31 December 2009, using generalised linear models with the Poisson regression link, and stratifying the analysis by the presence or absence of biomass advection due to combustion from wildfires. In addition to the above-described air pollution variables and heat-wave temperatures, in these models we also controlled for trend of the series, day of the week, annual, six-monthly and quarterly seasonalities, and the autoregressive nature of the series.

In view of the fact that the effect of PM and other control variables may be lagged in time, we created lags for the pollution variables up to order 4 for pollutants and heat (Díaz et al., 2015). Variables were selected using the Step-Step procedure, eliminating those that were not significant at $p < 0.05$. Models were constructed for DNBA and DBA, on the basis of data furnished by the *MAPAMA* for days with (DBA = 1) and without advections (DBA = 0).

Poisson modelling makes it possible to ascertain the RRs of mortality in respect of those variables which prove statistically significant. Based on these RRs, we calculated the corresponding increase in risk (IRR)

for every 10-μg/m^3 increase in PM_{10} concentrations. The analysis was performed on DBA and DNBA, for the complete period, the summer (June–September), and the remainder of the year (excluding summer).

Given that the synoptic-scale meteorological characteristics present in wildfires also favour Saharan dust intrusions (Díaz et al., 2017), the latter were excluded from the DBA analysis, so as to differentiate between the respective effects of the two types of PM advections on daily mortality.

3. Results

No DBA were detected in the Canary Islands and only one DBA was identified in the North region during the study period. It should be stressed that DBA frequently occurred at the same time as Saharan dust days, with the simultaneous development of both events accounting for 13% to 100% of all DBA (Table 1). The separate occurrence of DBA and Saharan dust advections caused statistically significant increases in regional background PM_{10} levels in each study area on mainland Spain, with the exception of the North and North-east regions for DBA. Hence, the highest increases in regional background PM_{10} levels were registered in all areas except the North on days when both types of events took place (Table 1). Atmospheric pollutant concentrations registered at hot-spots in urban sites can be divided into three different types, i.e., regional background contributions, urban background contributions, and contributions from specific sources in the proximity of the hot-spot, such as local traffic (Salvador et al., 2015). Owing to the increase in regional background PM_{10} levels in 7 of the 9 main study areas during DBA, PM_{10} levels registered at most of the urban sites in these areas could also be expected to increase during the occurrence of wildfire events.

Table 2 shows daily mean PM_{10} concentrations on DBA versus DNBA at a provincial capital level. Of the total of 34 provincial capitals in which DBA were detected, PM_{10} levels were higher in 28 (82.3%), with statistically significant differences ($p < 0.05$) for DBA versus DNBA.

Table 3 shows these PM_{10} concentrations averaged for all the provincial capitals that make up each geographical region used in this study, as well as the number of days on which there were DBA and DNBA in each region. In 6 of the 9 regions, these differences proved statistically significant. In terms of the frequency of appearance of DBA across the study period, this value ranged from zero in the Canary Islands region to 186 days (8.5%) in the North-west region.

To take into account the marked seasonal component of DBA in the summer months, we drew up Table 4 showing the frequency of appearance and concentrations of PM_{10} in the different regions, broken down by summer months and the remainder of the year. As will be seen, the maximum frequency of appearance of DBA took place in the summer as opposed to the remaining months of the year. In the summer months, in all regions where there were DBA (a total of 6), PM_{10} concentrations were higher on DBA than on DNBA, and in 5 of these regions these differences proved statistically significant.

Table 2
PM_{10} concentrations (μg/m^3) in the different provincial capitals on days with biomass advections (DBA) versus days without biomass advections (DNBA).

	PM_{10}			
	Days with biomass		Days without biomass	
	Mean	95% CI	Mean	95% CI
Albacete[a]	64.0	(60.17–67.91)	42.3	(41.49–43.04)
Alicante	–		–	
Almería	51.0	(35.75–66.25)	36.6	(34.94–38.18)
Avila	–		–	
Badajoz[a]	35.5	(29.98–41.05)	20.8	(20.33–21.21)
Barcelona	–		–	
Bilbao[a]	17.0	(17–17)	34.5	(33.76–35.23)
Burgos[a]	33.4	(30.7–36.06)	26.3	(25.8–26.78)
Cáceres[a]	26.2	(23.37–29.02)	18.2	(17.8–18.62)
Cadiz	–		–	
Castellón	–		–	
Ciudad Real	–		–	
Cordoba[a]	66.0	(58.02–74.04)	39.9	(38.98–40.78)
Corunna (*Coruña*)[a]	43.2	(37.35–48.98)	33.1	(31.92–34.22)
Cuenca	44.8	(31.74–57.76)	30.8	(29.59–32.03)
Gerona	–		–	
Granada	48.5	(41.11–55.96)	42.2	(41.22–43.12)
Guadalajara[a]	43.4	(38.88–47.93)	28.9	(28.17–29.72)
Huelva[a]	56.6	(47.94–65.28)	33.5	(32.84–34.1)
Huesca	–		–	
Jaén[a]	53.3	(44.34–62.34)	40.0	(39–40.99)
Las Palmas GC	–		38.1	(36.62–39.66)
León[a]	40.3	(38.33–42.31)	35.9	(35.21–36.61)
Lerida	–		–	
Logroño[a]	38.9	(34.49–43.31)	27.1	(26.51–27.73)
Lugo	–		–	
Madrid[a]	44.2	(40.35–48.02)	31.1	(30.45–31.79)
Malaga[a]	45.3	(33.95–56.66)	29.7	(28.9–30.53)
Murcia[a]	39.3	(35.07–43.6)	29.3	(28.57–30.1)
Orense	26.9	(21.45–32.36)	21.5	(20.5–22.54)
Oviedo[a]	48.5	(48.5–48.5)	40.8	(40.05–41.65)
P Mallorca[a]	55.0	(29.52–80.48)	28.6	(28.09–29.16)
Pamplona[a]	20.8	(20.8–20.8)	31.4	(30.76–31.98)
Pontevedra	–		–	
Salamanca[a]	39.4	(33.02–45.74)	24.6	(24.09–25.03)
San Sebastián[a]	16.0	(16–16)	27.3	(26.71–27.84)
SC de Tenerife	–		62.2	(58.72–65.62)
Santander[a]	19.3	(19.3–19.3)	32.3	(31.78–32.77)
Segovia[a]	36.5	(32.76–40.29)	24.3	(23.6–24.99)
Seville[a]	57.6	(49.45–65.65)	40.3	(39.49–41.06)
Soria[a]	34.2	(32.13–36.36)	27.3	(26.67–27.83)
Tarragona	–		–	
Teruel	–		–	
Toledo[a]	46.5	(41.56–51.37)	36.3	(35.52–37.1)
Valencia	32.4	(28.17–36.59)	29.5	(28.94–30.14)
Valladolid[a]	25.9	(22.65–29.23)	16.9	(16.5–17.3)
Vitoria	10.0	(10–10)	24.9	(24.31–25.57)
Zamora[a]	33.5	(30.17–36.74)	29.5	(28.97–30.07)
Zaragoza[a]	64.0	(47.56–80.53)	42.6	(41.73–43.55)

[a] Statistically significant differences at $p < 0.05$ between PM_{10} concentrations on DBA and DNBA.

Table 1
Mean PM_{10} concentrations (μg/m^3) at regional background monitoring sites by regions and days with biomass advections, with Saharan dust advections, with both types of events, and other days without biomass and Saharan dust advections.

Region	Days with biomass advections		Days with Saharan dust		Days with Saharan dust and biomass advections		Other days	
	Total days	Mean PM_{10} (95% CI)	Total days	Mean PM_{10} (95% CI)	Total days	Mean PM_{10} (95% CI)	Total days	Mean PM_{10} (95% CI)
North-west	150	18.1 (16.5–19.8)[a]	188	19.3 (17.7–20.8)[a]	22	33.5 (26.3–40.8)[a]	1628	10.4 (10.1–10.8)
North	1	5.0	205	25.8 (24.0–27.5)[a]	0		1676	16.9 (16.5–17.4)
North-east	8	27.0 (11.5–42.5)	354	27.9 (26.3–29.5)[a]	9	47.9 (28.5–67.3)[a]	1798	13.4 (13.0–13.8)
Centre	27	18.5 (16.0–21.1)[a]	324	26.0 (23.8–28.3)[a]	28	37.3 (31.9–42.8)[a]	1469	10.0 (9.7–10.3)
South-west	30	31.4 (27.1–35.6)[a]	518	24.4 (23.0–25.8)[a]	30	46.6 (31.1–62.0)[a]	1471	12.7 (12.3–13.0)
South-east	6	27.7 (18.0–37.3)[a]	653	33.6 (31.6–35.5)[a]	9	39.0 (30.4–47.6)[a]	1409	14.3 (13.9–14.7)
East	1	25.0*	474	25.3 (23.9–26.7)[a]	13	30.5 (23.9–37.2)[a]	1606	12.3 (11.9–12.6)
Balearic Isles	0		472	34.1 (32.5–35.8)[a]	4	44.8 (36.4–53.1)[a]	1517	18.6 (18.3–19.0)

[a] Statistically significant differences at $p < 0.05$ between days with biomass and/or Saharan dust advections and other days.

Table 3
Frequency of appearance of days with biomass advections (DBA) and days without biomass advections (DNBA) and PM_{10}concentrations (μg/m^3), broken down by the 9 regions considered.

Region	Days with biomass advections			Days without biomass advections		
	Total days	Mean PM_{10} (95% CI)	Mean $PM_{2.5}$ (95% CI)	Total days	Mean PM_{10} (95% CI)	Mean $PM_{2.5}$ (95% CI)
North-west*	186	38.92 (37.04–40.8)		2006	32.33 (31.78–32.88)	
North	1	21.93 (11.09–32.77)		2191	31.85 (31.58–32.12)	
North-east*	17	64.05 (47.57–80.53)		2175	42.64 (41.74–43.54)	
Centre*	56	38.92 (37.57–40.27)	23.57 (21.53–25.61)	2136	27.76 (27.56–27.96)	16.95 (16.62–17.28)
South-west*	60	54.16 (50.06–58.26)		2132	33.73 (33.34–34.12)	
South-east*	16	48.09 (43.54–52.64)		2176	36.19 (35.72–36.66)	
East	14	32.38 (28.17–36.59)		2178	29.54 (28.93–30.15)	
Balearic Isles	2*	55.00 (29.52–80.48)		2190	28.63 (28.10–29.16)	
Canary Islands	0			2192	46.21 (44.62–47.8)	14.14 (13.71–14.57)

* Statistically significant differences at $p < 0.05$ between PM_{10} concentrations on DBA and DNBA.

Daily natural-cause mortality values by province are shown in Table 5. As a general rule, in provinces that experienced days with biomass advections, mean daily mortality values were lower on DBA than on DNBA, with this difference being statistically significant in 17 out of a total of 47 provincial capitals. This same finding was in evidence when daily mortality was grouped by region, as can be seen from Table 6, in which daily mortality in all regions is shown as being lower on DBA than on DNBA.

The results of Poisson modelling for the complete period are listed in Table 7. Three differentiated behaviour patterns can be discerned: firstly, the North and South-east regions, where there was no association with mortality on DBA, while on DNBA such an association was evident; secondly, the North-west and South-west regions, where there was an effect on mortality on DBA, or this effect was reinforced; and thirdly, regions with $PM_{2.5}$ measures, where there was a shift in the mortality pattern from PM_{10} on DBA to $PM_{2.5}$ on DNBA. Furthermore, the impact had by PM_{10} on mortality, measured in IRR terms, was very much greater, with statistically significant differences, on DBA versus DNBA in regions with the above-described differentiated behaviour patterns.

If the analysis is performed exclusively for the summer months (Table 8), during which DBA are most frequent, no effect is seen in the North-west or South-east regions and there is no shift in the PM_{10}-$PM_{2.5}$ pattern in the Centre region. These results for DBA did not vary in the results for the whole year, regardless of whether or not there were Saharan dust intrusions (Table 9). Comparisons were made with the models corresponding to the summer months rather than the whole year, since it is during the summer that Saharan dust intrusions tend to occur most frequently.

4. Discussion

The results in Table 2 indicate that DBA give rise to a statistically significant increase in PM_{10} concentrations in most of the provinces analysed. This is due to the increase in regional background PM_{10} levels when DBA occur. It should be noted that, when DBA and Saharan dust advections took place simultaneously, the increase in the regional background PM_{10} levels was highest in all regions except the North. In this latter area, the occurrence of DBA was very rare, with no significant impact on regional background PM_{10} levels. Hence, the impact of burning biomass particles on population health in this region's urban areas could be expected to be negligible.

Analysis by region yielded similar results in terms of increases in PM_{10} on DBA versus DNBA. Perhaps the most notable finding is that it is in the North-west region where most DBA occur (186 days). This finding is in line with the number of wildfires that occurred in Spain across the period 2001–2010 (MAPAMA, 2012), with Galicia, which corresponds to the North-west region, being the territory where 42.5% of all Spain's wildfires occur. Nonetheless, the largest percentage of surface

Table 4
Frequency of appearance of days with biomass advections (DBA) and days without biomass advections (DNBA) and PM_{10} concentrations (μg/m^3), broken down by the 9 regions considered, according to different times/seasons of the year.

Region	Days with biomass advections			Days without biomass advections		
	Total days	Mean PM_{10} (95% CI)	Mean $PM_{2.5}$ (95% CI)	Total days	Mean PM_{10} (95% CI)	Mean $PM_{2.5}$ (95% CI)
Summer months (June–September)						
North-west[a]	120	35.42 (33.43–37.42)		612	26.88 (26.19–27.57)	
North	0	–		732	31.05 (30.64–31.46)	
North-east[a]	12	73.06 (53.83–92.29)		720	44.53 (43.09–45.96)	
Centre[a]	43	37.53 (36.06–39)	22.35 (20.08–24.62)	689	29.2 (28.85–29.55)	17.22 (16.72–17.73)
South-west[a]	43	55.55 (50.54–60.56)		689	37.18 (36.5–37.85)	
South-east[a]	10	51.81 (46.16–57.47)		722	39.4 (38.69–40.1)	
East	13	30.49 (28.3–32.68)		719	29.01 (28.14–29.87)	
Balearic Isles	0	–		732	29.75 (28.91–30.59)	
Canary Islands	0	–		732	43.13 (41.63–44.64)	13.12 (12.72–13.52)
Rest of the year						
North-west[a]	66	43.1 (39.86–46.34)		1394	34.9 (34.18–35.62)	
North	1	21.93 (11.1–32.76)		1459	32.26 (31.89–32.62)	
North-east	5	42.42 (17.52–67.31)		1455	41.71 (40.55–42.87)	
Centre[a]	13	43.78 (40.59–46.97)	27.63 (23.6–31.65)	1447	27.07 (26.84–27.3)	16.82 (16.4–17.24)
South-west[a]	17	50.71 (43.76–57.66)		1443	32.11 (31.63–32.59)	
South-east[a]	6	42.5 (35.33–49.67)		1454	34.61 (34–35.22)	
East[a]	1	55 (55–55)		1459	29.78 (29–30.57)	
Balearic Isles[a]	2	55 (29.52–80.48)		1458	28.07 (27.4–28.75)	
Canary Islands	0	–		1460	47.77 (45.51–50.04)	14.65 (14.03–15.27)

[a] Statistically significant differences at $p < 0.05$ between PM_{10} concentrations on DBA and DNBA.

area affected by major fires was not in North-west but in Andalusia (South-west and South-east regions), with 58.94% of its surface area affected by major fires (MAGRAMA, 2012). These data go to indicate that, though there are many fires in the North-west region, they are of a smaller magnitude than those in the South-west and South-east, where the fires are proportionally fewer but of greater magnitude in terms of hectares (ha) affected, especially in the summer months when DBA are most frequent, as can be seen in Table 4.

Tables 5 and 6 indicate, both at a provincial level and at the level of the different regions analysed respectively that mean daily mortality is higher on DNBA than on DBA. This finding does not run counter to our hypothesis to the effect that DBA produce an association between mortality and PM_{10} which is not seen on the remaining days. The natural-cause mortality analysed in this study is attributable to many more causes than just the circulatory and respiratory causes which, as stated in the Introduction, may be DBA-related and be slightly increased on these days. In the period analysed, respiratory- and circulatory-cause mortality accounted for 11.8% and 34.8% of natural-cause mortality respectively (Ortiz et al., 2017). It is precisely this which lends greater relevance to the Poisson modelling performed in this study, capable of detecting such non-evident associations.

The pattern observed in Table 7 with respect to the association between PM_{10} and daily mortality coincides with regions having a higher frequency of DBA, and indeed the three regions that display significant associations were those with the highest DBA frequency. The negligible number of DBA in the remaining regions might account for the fact that significant associations between PM_{10} and daily mortality were detected, not on DBA, but on the rest of the days (DNBA), with the particulate matter in question being mainly of anthropic origin (Ortiz et al., 2017). This result is also in line with the shift in the PM-mortality pattern observed for the Centre region, with vehicle-borne $PM_{2.5}$ being the particles linked to mortality on DNBA (Linares et al., 2015).

Furthermore, observation shows that, while the effect of PM_{10} on mortality tends to be somewhat more lagged (lags 2–4) on DBA than on DNBA (lags 1–2), the impact of PM_{10} on mortality is much greater on DBA than on DNBA. The reason for this different pattern is likely to be found in the different toxicological nature of PM_{10} in the two cases (Finlay et al., 2012). The difference between urban and wildfire smoke was also illustrated in a study which examined the effect on macrophages exposed to wildfire and urban smoke in a murine model (Jalava et al., 2006). This showed that, though cytokine production in response to wildfire smoke was lower than with urban-derived particles, there was increased inflammatory (determined by measuring proinflamatory cytokines) and cytotoxic activity (as measured by biochemical markers of toxicity, apoptotic and nitrous oxide production) per cubic metre of air containing wildfire particles than with air containing only urban particulate matter. Presumably, this was as a result of a higher concentration of PM_{10} in the wildfire smoke. This increased particulate size means that particles can accumulate in the lung more easily. Furthermore, the volatile organic compounds present in wildfires (Garcia-Hurtado et al., 2013) could be absorbed into the surface of the particles (Finlay et al., 2012), thereby boosting their toxicity.

Another factor which might account for the higher IRR on DBA than on DNBA are the high temperatures and high ozone levels found in the majority of wildfires (Jaffe et al., 2013), and the synergic effect which high temperatures have on particle behaviour. In addition, there are the many interactions, on the one hand between heat and particulate matter - highlighted in many multicentre and case-crossover studies (Sartor et al., 1995; Stafoggia et al., 2010; Stieb et al., 2009) - and on the other, between heat and ozone (Sartor et al., 1995; Díaz et al., 2002).

One result that might, in principle, strike one as surprising is the fact that no association between mean daily mortality and PM_{10} was observed in the North-west region for the summer months, when such an association was observed for the year as a whole. The North-west is

Table 5
Mean daily natural-cause mortality in the different provinces on days with biomass advections (DBA) versus days without biomass advections (DNBA). In the table appears the name of the capital's province.

Town	Natural-cause mortality			
	Days with biomass		Days without biomass	
	Mean	95% CI	Mean	95% CI
Albacete	8.6	(7.9–9.31)	9.1	(8.92–9.19)
Alicante	36.9	(33.82–39.89)	36,0	(35.65–36.3)
Almería	11.4	(9.15–13.6)	11.6	(11.47–11.79)
Avila	5.1	(4.58–5.67)	4.9	(4.76–4.95)
Badajoz[a]	14.9	(13.97–15.9)	17.2	(16.97–17.38)
Barcelona	112.1	(106.45–117.67)	116.1	(115.21–116.91)
Bilbao	28,0	(28–28)	28.1	(27.8–28.35)
Burgos	8.5	(7.81–9.23)	9.2	(9.03–9.3)
Cáceres	9.9	(8.89–10.82)	10.4	(10.25–10.56)
Cadiz[a]	20.8	(19.54–22.13)	23,0	(22.72–23.24)
Castellón	11.1	(9.67–12.47)	12.4	(12.28–12.61)
Ciudad Real	12.8	(11.96–13.57)	13.3	(13.17–13.52)
Cordoba	17.2	(16.13–18.27)	18.9	(18.65–19.11)
Corunna (*Coruña*)[a]	30.5	(29.47–31.47)	32.1	(31.76–32.36)
Cuenca	4.9	(4.27–5.44)	4.9	(4.83–5.02)
Gerona[a]	12.6	(10.93–14.36)	14.9	(14.67–15.03)
Granada	19.6	(17.31–21.82)	20.2	(19.97–20.43)
Guadalajara	4.4	(3.76–5.03)	4.6	(4.48–4.67)
Huelva	11.1	(10.17–12.09)	11,0	(10.84–11.16)
Huesca	5.9	(4.86–7.02)	5.9	(5.82–6.04)
Jaén[a]	12.8	(10.61–14.89)	15.3	(15.11–15.49)
Las Palmas de GC	–		16.6	(16.42–16.82)
León	14.3	(13.67–14.86)	14.8	(14.66–15.03)
Lerida	9.4	(7.74–11.08)	10.7	(10.52–10.83)
Logroño	6.6	(6.01–7.17)	7.2	(7.09–7.33)
Lugo[a]	11.5	(10.98–12.04)	12.2	(12.06–12.4)
Madrid[a,b]	53,0	(50.64–55.36)	59.9	(59.43–60.35)
Malaga	31.1	(28.33–33.92)	30.9	(30.63–31.21)
Murcia	24.8	(21.79–27.71)	26.3	(26.01–26.53)
Orense[a]	11,0	(10.54–11.51)	11.7	(11.56–11.9)
Oviedo[a]	24,0	(24–24)	33.1	(32.83–33.45)
P. Mallorca[a]	15,0	(11.08–18.92)	20.4	(20.18–20.61)
Pamplona[a]	11,0	(11–11)	13.6	(13.38–13.72)
Pontevedra	21,0	(20.2–21.81)	21.3	(21.11–21.58)
Salamanca	9.3	(8.4–10.1)	10.2	(10.01–10.29)
San Sebastián[a]	14,0	(14–14)	16.1	(15.87–16.27)
SC de Tenerife	–		17.4	(17.23–17.64)
Santander	19,0	(19–19)	14.6	(14.42–14.78)
Segovia	4,0	(3.41–4.59)	4,0	(3.96–4.14)
Seville[a]	35.9	(34.16–37.68)	38.9	(38.49–39.23)
Soria	3,0	(2.57–3.39)	2.8	(2.72–2.86)
Tarragona[a]	13.7	(11.28–16.13)	16.3	(16.15–16.52)
Teruel	4.2	(3.05–5.42)	4.2	(4.11–4.28)
Toledo	12.9	(11.8–13.98)	13.9	(13.68–14.04)
Valencia[a]	49.9	(46.38–53.33)	57.4	(56.96–57.89)
Valladolid	11.6	(10.48–12.67)	12.6	(12.43–12.76)
Vitoria[a]	8,0	(8–8)	6.1	(6.02–6.25)
Zamora	6.3	(5.55–6.98)	6.1	(6.04–6.25)
Zaragoza[a]	22.3	(20.4–24.19)	24.5	(24.26–24.76)

[a] Statistically significant differences at $p < 0.05$ between mean daily mortality on DBA and DNBA.
[b] Daily mortality only in metropolitan city.

Table 6
Mean daily mortality broken down by the 9 regions considered.

Region	Days with biomass advections	Days without biomass advections
	Mean natural-cause (95% CI)	Mean natural-cause (95% CI)
North-west*	17.66 (17.09–18.22)	18.44 (18.26–18.62)
North	17.33 (11.14–23.52)	18.60 (18.42–18.78)
North-east	25.76 (19.25–32.26)	27.51 (26.92–28.11)
Centre	10.71 (9.88–11.54)	11.53 (11.38–11.69)
South-west*	20.00 (18.89–21.12)	21.78 (21.57–21.99)
South-east	19.91 (17.95–21.87)	20.86 (20.70–21.03)
East	32.60 (27.42–37.77)	35.28 (34.79–35.77)
Balearic Isles*	15.00 (11.08–18.92)	20.39 (20.18–20.61)
Canary Islands	–	17.03 (16.88–17.17)

* Statistically significant differences at $p < 0.05$ between mean daily mortality on DBA and DNBA.

Table 7
PM variables that proved significant in the Poisson regression models fitted for each provincial capital of the respective regions considered, for the whole year.

Region	Capital	Days with biomass advections			Days without biomass advections		
			RR (95% CI)	IRR[a] (%) (95% CI)		RR (95% CI)	IRR[a] (%) (95% CI)
North	Bilbao		–		PM_{10} (lag 1)	1.006 (1.001–1.011)	0.63 (0.12–1.13)
North-east	Zaragoza		–			–	
North-west	Corunna	PM_{10} (lag 4)	1.079 (1.024–1.138)	7.93 (2.36–13.81)		–	
Centre	(*Coruña*) Madrid	PM_{10} (lag 2)	1.038 (1.014–1.062)	3.76 (1.36–6.22)	$PM_{2.5}$ (lag 2)	1.010 (1.002–1.018)	0.98 (0.17–1.80)
South-east	Malaga		–		PM_{10} (lag 1)	1.006 (1.001–1.010)	0.57 (0.14–0.99)
South-west	Seville	PM_{10} (lag 0, 2)	1.045 (1.030–1.059)	4.46 (2.99–5.94)	PM_{10} (lag 0)	1.005 (1.001–1.009)	0.48 (0.07–0.88)
East	Valencia		–			–	
Canary Islands	Las Palmas G.C.		–			–	
Balearic Isles	Palma de Mallorca		–			–	

RR: relative risk.
IRR: increase in relative risk.
[a] For these chemical air-pollutant variables increases of 10 μg/m^3.

where most DBA occur, fundamentally in summer. The reason is probably to be found in what was said above, namely, that in the North-west there are more wildfires, though of smaller proportions, than in the South-west. These findings would seem to indicate that it is the fires of large dimensions which really have repercussions on mortality. This coincides with a study conducted in the city of Athens (Analitis et al., 2012), which reported that fires with an affected surface area of <5 ha had no effect on mortality, and that the larger the affected surface area, the greater the impact on mortality. It would therefore seem logical that the effect on mortality due to PM advections from biomass combustion in the North-west, where the mean fire-affected area is only 4 ha, would not be statistically significant as compared to the South-west, where the mean fire-affected area is in excess of 8 ha (MAGRAMA, 2012).

In some regions, the synoptic-scale atmospheric situations which produce DBA tend to coincide with those which produce other advections of natural origin, such as Saharan dust (Díaz et al., 2017). It thus becomes a matter of ascertaining whether these two effects have independent results on daily mortality. The results in Table 9 indicate that, if one eliminates those DBA on which there are also Saharan dust intrusions, the pattern of association between PM_{10} and mean daily mortality remains the same, i.e., the pattern is present in the same regions as shown in Table 8. In quantitative terms, however, as measured by reference to the RRs for PM_{10} and daily mortality, the effect of PM_{10} on DBA is in fact reduced if the effect of this natural advection is eliminated.

A limitation to our work is that, we had no explanatory variables, a part from sex, age and the address of the subjects, at the individual level. In particular, we cannot control for factors, such as individual socioeconomic data, lifestyles and comorbidities that may differ between the mortality in people in different cities (Vodonos et al., 2015). These factors may also act as confounders or effect modifiers of relations between air pollution and daily mortality (Barceló et al., 2016). It should likewise be noted that, as this was a longitudinal ecological study, the results cannot be extrapolated at an individual level. The exposure levels used were based on exposures determined on the basis of readings taken by external monitors and then averaged, with the result that they are not measures which represent individual exposure. Another bias from the monitors used can be the heterogeinity in the type, they are mostly urban but occasionally we used background type. Even so, this is a commonly used methodology in these types of studies (Samet et al., 2000). However, much of this residual confusion is controlled by inclusion in the model of variables such as: trend of the series, day of the week, annual, six-monthly and quarterly seasonalities and the autoregressive nature of the series.

At all events, most air-pollution studies address the misalignment problem (albeit only implicitly), by using a two-stage modelling procedure, or plug-in approach, where predictions from an exposure model (first stage) are used as covariates in a health-effect model (second stage) (Barceló et al., 2016). Although in a few cases, predictions are obtained from exposure models that explicitly incorporate spatial structure, even in such cases the plug-in approach does not take into account the uncertainty in the exposure predictions. This leads to a complex form of measurement error, resulting in bias of the health effect (Wannemuehler et al., 2009; Ingebrigtsen et al., 2015).

With reference to the previous papers published on this topic and outlined in the Introduction, our study is unique in that it covers all the regions of a single country, with similar demographic and socio-economic conditions prevailing in all the regions analysed. Socio-economic and demographic differences can bias the end results obtained on comparing the findings for different cities in different countries (Faustini et al., 2015). A study undertaken in Europe (Temam et al., 2017) highlights the fact that equal distribution to air pollution exposure across socio-economic groups is complex in the context of European cities, and that, rather than there being a general city-wide pattern, inequalities

Table 8
PM variables that proved significant in the Poisson regression models fitted for each provincial capital in the respective regions considered, in summer months (June–September).

Summer months (June–September)		Days with biomass advections			Days without biomass advections		
Region	Capital		RR (95% CI)	IRR[a] (%) (95% CI)		RR (95% CI)	IRR[a] (%) (95% CI)
North	Bilbao		–		PM_{10} (lag 1)	1.023 (1.013–1.033)	2.30 (1.33–3.29)
North-east	Zaragoza		–			–	
North-west	Corunna (Corunna (Coruña))		–			–	
Centre	Madrid	PM_{10} (lag 2)	1.058 (1.024–1.093)	5.81 (2.39–9.35)		–	
South-east	Malaga		–			–	
South-west	Seville	PM_{10} (lag 2)	1.033 (1.019–1.047)	3.32 (1.93–4.74)	PM_{10} (lag 0)	1.016 (1.008–1.024)	1.59 (0.76–2.43)
East	Valencia		–			–	
Canary Islands	Las Palmas G.C.		–			–	
Balearic Isles	Palma de Mallorca		–			–	

RR: relative risk.
IRR: increase in relative risk.
[a] For these chemical air-pollutant variables, increases of 10 μg/m^3.

Table 9
PM variables that proved significant in the Poisson regression models fitted for each provincial capital in the respective regions considered, for the whole year.

Region	Capital	Days with biomass advections but without Saharan dust advections			
		Total days		RR (95% CI)	IRR[a] (%) (95% CI)
North	Bilbao	1		–	
North-east	Zaragoza	8		–	
North-west	Corunna (*Coruña*)	163		–	
Centre	Madrid	37	PM_{10} (lag 2)[b]	1.034 (0.999–1.070)	3.41 (−0.08–7.02)
South-east	Malaga	7		–	
South-west	Seville	32	PM_{10} (lag 0)	1.027 (1.000–1.054)	2.67 (0.02–5.39)
East	Valencia	2		–	
Canary Islands	Las Palmas G.C.	0		–	
Balearic Isles	Palma de Mallorca	0		–	

RR: relative risk.
IRR: increase in relative risk.
[a] For these chemical air-pollutant variables, increases of 10 μg/m^3.
[b] Significant at a p-value < 0.060.

need to be specifically assessed in each city. Hence, the association reported by Hajat in the USA is not so clear in the case of Europe, i.e., that lower socio-economic or minority groups are more likely to be exposed to higher traffic-related air-pollution exposure in the form of nitrogen dioxide (NO_2) and/or PM (Hajat et al., 2015).

From the low number of days in Spain on which there are PM advections from biomass combustion, one might be led to think that, from a public health standpoint, the relevance of such advections is small. There are, however, several reasons for contradicting such a conclusion: firstly, since PM_{10} poses a higher RR of mortality on DBA than on DNBA, and a high number of persons are exposed when the situation is viewed from a regional perspective, the effect on mortality is important; secondly, this is an effect which comes *in addition to* the effects of the chemical pollution that already exists in major cities and the high temperatures associated with a large proportion of wildfires; and lastly, there is the matter of future perspectives, since, according to the IPCC report (IPCC, 2013), wildfires in Mediterranean Europe are set to increase, both in frequency and in the amount of hectares affected, making this type of study crucial from the point of view of implementing measures designed to minimise their impact on health. From the results obtained we can conclude that, the increase in the PM concentrations caused by biomass advections from wildfires is linked to a significant IRR of mortality in Spain. This increase affects to the exposure of people to PM, so groups of especially vulnerability such as children, elderly or people with cardiorespiratory diseases, can be importantly damaged. From a point of view of Public Health, informative measures have to be implemented to alert of this risk when this intrusions occurs.

Disclaimer

This paper reports independent results and research. The views expressed are those of the authors and not necessarily those of the Carlos III Institute of Health.

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